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A General Framework for Setting Quantitative Population Objectives for Wildlife Conservation

Kristen E. Dybala^{*1}, Neil Clipperton², Thomas Gardali¹, Gregory H. Golet³, Rodd Kelsey⁴, Stefan Lorenzato⁵, Ron Melcer, Jr.^{5,6}, Nathaniel E. Seavy¹, Joseph G. Silveira⁷, and Gregory S. Yarris⁸

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* Corresponding author: kdybala@pointblue.org

- 1 Point Blue Conservation Science
Petaluma, CA 94954 USA
- 2 California Department of Fish and Wildlife
Sacramento, CA 95811 USA
- 3 The Nature Conservancy
Chico, CA 95926 USA
- 4 The Nature Conservancy
Sacramento, CA 95814 USA
- 5 California Department of Water Resources
Sacramento, CA 95814 USA
- 6 Geography Graduate Group, University of California
Davis, CA 95616 USA
- 7 U.S. Fish and Wildlife Service
Sacramento National Wildlife Refuge Complex
Willows, CA 95988 USA
- 8 Central Valley Joint Venture, U.S. Fish and Wildlife Service
Sacramento, CA 95825 USA

ABSTRACT

Quantitative population objectives are necessary to successfully achieve conservation goals of secure or robust wildlife populations. However, existing methods for setting quantitative population objectives commonly require extensive species-specific population viability data, which are often unavailable, or are based on estimates of historical population sizes, which may no longer represent

feasible objectives. Conservation practitioners require an alternative, science-based method for setting long-term quantitative population objectives. We reviewed conservation biology literature to develop a general conceptual framework that represents conservation biology principles and identifies key milestones a population would be expected to pass in the process of becoming a recovered or robust population. We then synthesized recent research to propose general hypotheses for the orders of magnitude at which most populations would be expected to reach each milestone. The framework is structured as a hierarchy of four population sizes, ranging from *very small* populations at increased risk of inbreeding depression and extirpation (<1,000 adults) to *large* populations with minimized risk of extirpation (>50,000 adults), along with additional modifiers describing *steeply declining* and *resilient* populations. We also discuss the temporal and geographic scales at which this framework should be applied. To illustrate the application of this framework to conservation planning, we outline our use of the framework to set long-term population objectives for a multi-species regional conservation plan, and discuss additional considerations in applying this framework to other systems. This general framework provides a transparent, science-based method by which conservation practitioners and stakeholders can agree on long-term population objectives of an appropriate magnitude, particularly when

the alternative approaches are not feasible. With initial population objectives determined, long-term conservation planning and implementation can get underway, while further refinement of the objectives still remains possible as the population's response to conservation effort is monitored and new data become available.

KEY WORDS

Conservation plan, conservation objectives, population viability, recovery plan, resilience

INTRODUCTION

Conservation objectives are the specific, measurable changes that are necessary to achieve broad and visionary conservation goals (CMP 2013), such as improving ecological integrity (LCC 2014); enhancing ecosystem function, services, or resilience (UNEP 2010); and achieving recovered, secure, or robust wildlife populations (Rich et al. 2004; NOAA 2012). Quantitative conservation objectives determine when conservation goals have been reached, and measure progress toward these goals (Nicholson and Possingham 2006). Thus, generating clear and scientifically defensible conservation objectives is a critical component of successful conservation planning and implementation (Margules and Pressey 2000; Villard and Jonsson 2009). However, the process of establishing objectives is not straightforward and is subject to debate (Tear et al. 2005; Wilhere 2008).

When the conservation goal is to achieve recovered, secure, or robust wildlife populations, conservation objectives often take the form of a target population size. To establish these population objectives, one recommended approach is to estimate the minimum viable population (MVP) size (Himes Boor 2014; Doak et al. 2015), defined as the smallest population size at which the population has a high probability of persisting for a desired length of time (Morris and Doak 2002). Yet accurate estimates of MVP require extensive long-term demographic data that are often not available (Wolf et al. 2015), and analysis of inadequate data can have misleading results (Chapman et al. 2001; Patterson and Murray 2008). Few recovery plans for at-risk species have actually used population viability analyses to set recovery

criteria (Morris et al. 2002; Neel et al. 2012), and there is growing recognition that successful species conservation requires population sizes well above the MVP (Soulé et al. 2003; Sanderson 2006; Redford et al. 2011).

A second approach is to set population objectives based on estimates of historical population sizes or trends (Sanderson 2006; McClenachan et al. 2012). For example, the North America Waterfowl Management Plan originally set an objective of returning North American waterfowl populations to the sizes observed in the 1970s (NAWMP 1986), and Partners in Flight set an objective of returning North American landbird watch list species to population sizes observed in the 1960s (Rich et al. 2004). These objectives are designed to reverse population declines, guiding conservation plans toward conditions at a time when populations were likely more robust. Though historical population sizes can provide important context for understanding the magnitude of population declines (McClenachan et al. 2012), the time-period selected is arbitrary. If historical estimates are too recent or population objectives are too modest, once they are achieved it may be difficult to maintain support for additional conservation efforts (Petrie et al. 2011). If historical estimates are instead from a more distant time-period, they may be impossible to achieve in landscapes that have changed significantly as a result of urbanization, climate change, or other factors (Marsh et al. 2005; Lawler 2009).

Much has been written about general considerations that should go into setting population objectives (Sanderson 2006; Redford et al. 2011; Westwood et al. 2014), but aside from using historical or MVP size estimates, there is little guidance for choosing appropriate numbers. Science-based guidelines for choosing appropriate population objectives are needed in cases when population viability analyses are not feasible, when historical population size estimates are unavailable or inappropriate, or when the long-term goal is beyond recovery from recent declines. For example, regional or continental conservation plans that seek to keep common species common require an efficient method of setting population objectives for multiple species. In the absence of such guidelines, uncertainty about how to proceed, and fear of failure, can lead to

delays in taking conservation action (Meek et al. 2015), while objectives that are instead driven by political considerations are often far lower than what is necessary to achieve conservation goals (Svancara et al. 2005; Noss et al. 2012). An ability to efficiently set long-term population objectives that are “in the ballpark” of what is necessary to achieve secure, robust populations would allow conservation planning to get underway while still allowing refinement of the objectives as additional data become available.

We examined the literature on setting population objectives and the relevant underlying principles of conservation biology to develop a general population objective-setting framework that could be easily adapted to many taxa. We then make specific recommendations for population objectives and the spatial scale at which they should be established. To illustrate the application of this framework, we also describe the process we used to set population objectives for a suite of focal bird species, and discuss considerations in applying this framework to other systems.

CONCEPTUAL DEFINITION AND GENERALIZED FRAMEWORK

To develop a conceptual definition of a recovered or robust population, we examined recent conservation biology literature. Four features were commonly recommended: genetically robust, self-sustaining, ecologically functional (or effective), and resilient (Conner 1988; Soulé et al. 2003; Sanderson 2006; Redford et al. 2011; Wolf et al. 2015).

A *genetically robust* population is defined as sufficiently abundant that inbreeding and genetic drift are minimized, and with sufficient gene flow among sub-populations that genetic diversity is maintained, maximizing the capacity to adapt to changes in environmental conditions (Redford et al. 2011; Frankham et al. 2014). A *self-sustaining* population is both stable or growing in abundance and demographically viable, meaning it is sufficiently abundant that the risk of extinction or extirpation from demographic or environmental stochasticity is minimized (Traill et al. 2010; Redford et al. 2011). An *ecologically functional* population is sufficiently abundant that key ecological interactions and

functions are maintained (Conner 1988; Soulé et al. 2003), such as seed dispersal, nutrient transport, or its role in the food web (Peery et al. 2003; Estes et al. 2010; Galetti et al. 2013). Finally, a *resilient* population is one that can recover from disturbance, and is an important component of climate change adaptation strategies (Folke et al. 2004; Lawler 2009). A *resilient* population should comprise multiple genetically robust and self-sustaining sub-populations, encompassing the concepts of redundancy and representation of the genetic diversity across the species' range (Redford et al. 2011; Wolf et al. 2015). A *resilient* population can recover from an environmental catastrophe in one part of the range, and allows for shifts in distribution, which minimizes the risk of extinction or extirpation across the range.

Based on these conservation biology principles, we developed a generalized population status framework that describes the milestones a population would be expected to pass in the process of becoming a robust population (Table 1). The framework is structured as a hierarchy of four population sizes:

1. *Very small* populations at increased risk of inbreeding depression
2. *Small* populations with reduced risk of inbreeding depression but that are still vulnerable to long-term loss of genetic diversity and extinction or extirpation through demographic and environmental stochasticity
3. *Viable* populations with reduced vulnerability to extinction or extirpation, and
4. *Large* populations with minimized vulnerability to extinction or extirpation and increased ability to maintain key ecological interactions and functions.

We also identified two modifiers to these population status categories that reflect considerations beyond population size:

- *Steeply declining* populations that are at increased risk of extinction or extirpation no matter their population size (Stanton 2014), and
- *Resilient* populations with increased ability to recover from disturbances.

POPULATION SIZE RECOMMENDATIONS

For this general framework to be useful in setting population objectives, the criteria defining each population status must be quantified, and the current status of populations of interest must be estimated. Although a population objective can be set without knowledge of the current population size an estimate is necessary to measure progress toward achieving that population objective, and likely also to estimate the extent of conservation actions that will be required to achieve the objective. The growing number of methods for estimating abundance or relevant surrogates of abundance (e.g., Royle and Nichols 2003), along with the growing availability of large citizen science data sets (e.g., Sullivan et al. 2014; Barlow et al. 2015) and data repositories (Iliff et al. 2008; Michener et al. 2012), is making population size and trend estimation more feasible for many species. In contrast, species-specific genetics, demography, and population dynamics data that could help identify the sizes at which an individual population is expected to reach each population status are not available for most species, and the cost of obtaining such estimates may be prohibitive. Instead, we turned to the recent conservation biology literature to recommend numerical thresholds for each population status based on the best available information (Table 1).

For *very small* populations, inbreeding is known to have consistently deleterious effects on fitness across

taxa, and the well-known 50/500 rule has been used to identify the threshold at which populations are at increased risk of inbreeding depression (Mills et al. 2005). Based partly on observations of lab and domesticated animals, an effective population size of at least 50 breeding adults is estimated as necessary to avoid inbreeding depression in the short-term (Franklin 1980). Assuming a ratio of actual population size to effective population size of 10:1 (Frankham 1995), this corresponds to a total population size of 500. However, Frankham et al. (2014) have proposed a revision upward to an effective population size of 100 and a corresponding total population size of 1,000, in part because wild populations are likely to be more susceptible to inbreeding depression than domesticated or lab animals. Thus, we propose that population sizes of fewer than 1,000 be considered *very small* populations at increased risk of inbreeding depression and local extirpation.

Above this threshold for *very small* populations, *small* populations may still be vulnerable to loss of long-term genetic diversity and extirpation from demographic or environmental stochasticity. To maintain evolutionary potential over the long-term, the 50/500 rule also proposes that an effective population size of more than 500 breeding adults is needed (Franklin 1980). As above, Frankham et al. (2014) recommend revising this estimate upward to an effective population size of 1,000,

Table 1 Population status framework describing the milestones a population would pass in the recovery process, shown with proposed thresholds for the number of adult individuals in the population, additional modifiers, and numerical criteria

Population status	Description	Proposed thresholds
Very small	Expected to be well below minimum viable population size (MVP), and at increased risk of inbreeding depression in the short term.	< 1,000
Small	May be below MVP and vulnerable to extirpation through environmental and demographic stochasticity and long-term loss of genetic diversity.	< 10,000
Viable	Expected to meet or exceed MVP, reducing vulnerability to environmental and demographic stochasticity and preserving genetic diversity.	> 10,000
Large	Expected to be well above MVP, minimizing vulnerability to environmental and demographic stochasticity, preserving genetic diversity, and improving ability to maintain key ecological interactions and functions.	> 50,000
Additional modifiers		Criteria
Steeply declining	Increased risk of extinction or extirpation until the causes of the decline are addressed, no matter the population size.	> 30% decline in 10 years (observed or projected)
Resilient	Multiple viable or large populations to hedge against environmental catastrophes	At least two viable populations (> 10,000)

corresponding to a total population size of 10,000. Similarly, to minimize the risk of local extirpation from demographic or environmental stochasticity, studies have estimated average cross-taxa MVPs of a comparable magnitude. Reed et al. (2003) estimated MVPs for 102 vertebrate species, primarily birds and mammals, and identified a mean MVP of 7,316 (+/- 562) breeding adults to achieve a 99% probability of persisting 40 generations, with the frequency distribution peaking in the 4,900–8,100 range. Traill et al. (2007) conducted a meta-analysis of MVPs across 287 populations of 212 species, of which 47% were vertebrates, and identified a median MVP of 4,169 (95% CI: 3,577–5,129), with the frequency distribution peaking near 10,000 (Traill et al. 2007). Thus, from both a genetics and population dynamics standpoint, we propose that populations larger than 10,000 breeding adults are more likely to be above MVP and capable of persisting and maintaining genetic diversity over the long-term (i.e., more likely to be *viable* populations).

Generalizing MVP estimates across species has been criticized because the MVP for any individual population may vary widely with species and context, such as the variance in environmental conditions and the kinds of threats different species face (Brook et al. 2011; Flather et al. 2011). Yet, even among the critics of this approach, there is a general agreement that MVPs are likely to be in the thousands, not hundreds (Mills et al. 2005; Brook et al. 2011; Flather et al. 2011; Jamieson and Allendorf 2012). While we agree that there is no universal MVP threshold and that this remains an area of active research, we propose that 10,000 is a reasonable starting point for setting long-term population objectives intended to achieve long-term viability. This estimate is larger than the population size of 5,000 that has previously been recommended as a general rule of thumb, based roughly on median MVP estimates across taxa (Traill et al. 2010). However, given the finding that short-term data sets may under-estimate MVPs (Reed et al. 2003), the broader peak in the frequency distribution of MVP estimates (Reed et al. 2003; Traill et al. 2007), and the potential consequences of under-estimating the population size needed, we prefer to err on the side of over-estimating the population size necessary. Thus, we have recommended a population

size on the high side of the median MVP estimates (Noss et al. 2012).

Although meeting the MVP size is necessary for long-term persistence, populations should be well above MVP to maximize the likelihood that they are able to maintain key ecological interactions and functions (Conner 1988; Soulé et al. 2003; Sanderson 2006; Redford et al. 2011). Functional extinctions of species occur when species are not abundant enough to be ecologically effective, which can lead to cascading effects on ecosystems (Pyare and Berger 2003; Ripple et al. 2013), including declines of dependent species (Anderson et al. 2011), changes in evolutionary trajectory (Galetti et al. 2013), and phase shifts in ecosystem state (Estes et al. 2010). The point at which a population becomes ecologically effective is not well-studied, but, for example, the population size of salmon necessary to maintain its role in landscape nutrient dynamics was ten times the MVP (Peery et al. 2003). Depending on the ecological function in question, it may be that achieving a minimum density is more important than a specific population size, such as the minimum density of otters required to maintain kelp forests (Estes et al. 2010). Thus, other than “well above MVP,” there are no rules of thumb for ecological effectiveness yet available, and this, too, remains an area of active research.

In addition to providing for ecological function, populations should also be well above MVP to allow time for declines to be detected and reversed before a population falls below MVP. Here, we defined a *steeply declining* population based on the International Union for Conservation of Nature (IUCN) Red List criteria of an observed or projected decline of greater than 30% over 10 years or generation length, whichever is longer (Mace et al. 2008). Thus, a *steeply declining* population that starts at a population size of 15,000 could drop below 10,000 (i.e., the threshold for *viable*) before the *steeply declining* status is recognized. Indeed, Frankham et al. (2014) have suggested that declining populations of fewer than 20,000 adults should be considered vulnerable to extinction on the IUCN Red List. In lieu of better information about the population sizes or densities required to maintain key ecological functions, and in favor of over-estimating the population size necessary, we propose aiming for

large populations of more than 50,000 individuals, well above the threshold for *viable*, and providing time to detect and intervene in even precipitous population declines.

Finally, we defined a *resilient* population as having at least two *viable* or *large* sub-populations, presumably with at least some connectivity among them. Having more than one genetically robust, self-sustaining, and ecologically functional population improves representation of the full array of genetic diversity across the species range, and builds in redundancy, which hedges against disturbances or disasters in one part of the range (Redford et al. 2011; Wolf et al. 2015). Thus, to achieve a goal of supporting a *resilient*, genetically robust, and self-sustaining population, our framework provides a starting point for defining science-based population objectives that would require more than one sub-population, each of which is at least *viable* (>10,000 individuals), preferably *large* (> 50,000), and stable or increasing (i.e., not *steeply declining*).

GEOGRAPHIC AND TEMPORAL SCALES

This framework can be flexibly applied to define population objectives for any geographic or temporal scale. However, the appropriate scale for any individual case will depend on both the conservation goals and the biology of the species of interest. As examples, we discuss three cases: (Case 1) a species with a single population of limited distribution, (Case 2) a species with several discrete populations, and (Case 3) a wide-ranging species with a continuous distribution throughout its range. In any of these cases, the largest scale on which this framework should be applied is the spatial extent of a single biological population (Figure 1A), defined as a group of individuals of one species that is largely reproductively isolated such that changes in the population's size are driven primarily by births and deaths rather than movement (Berryman 2002). For example, for a conservation goal of achieving a genetically robust and self-sustaining population, a population objective of *viable* or *large* could be set

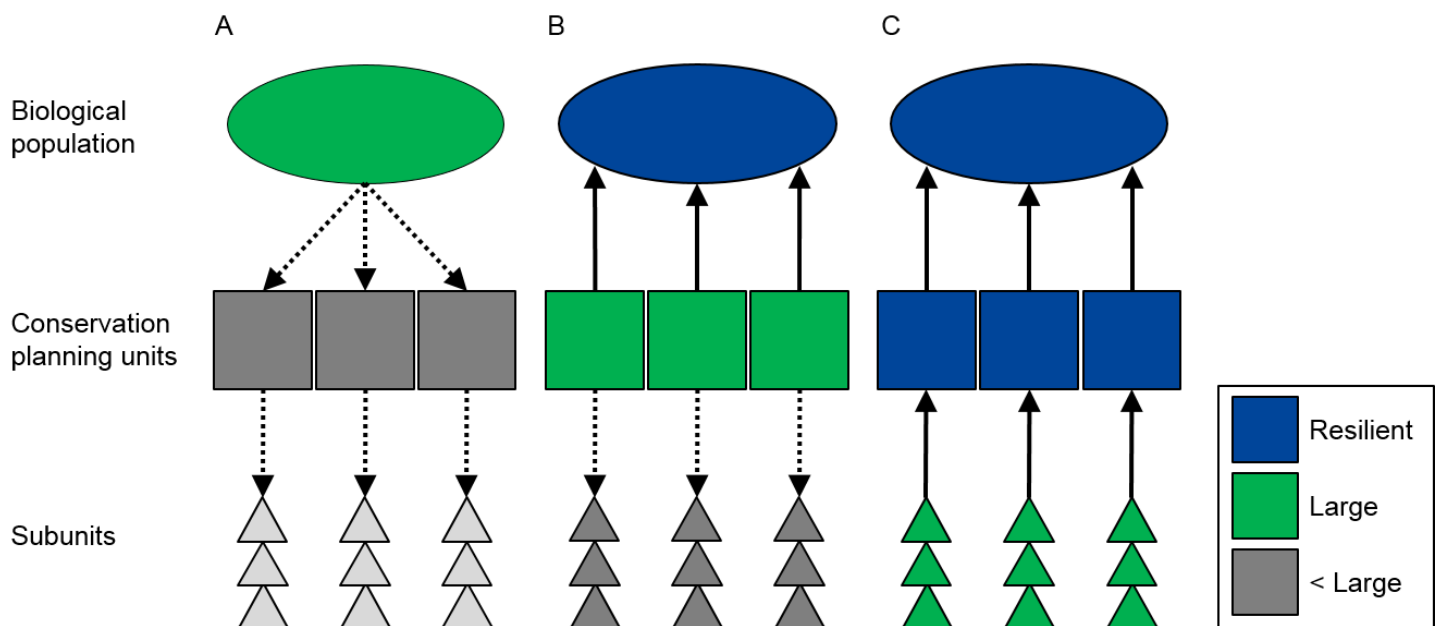


Figure 1 Conceptual diagram demonstrating the application of the population status framework to a wide-ranging species with a continuous distribution on different spatial scales: entire biological populations, conservation planning units within the population's range (such as ecoregions or joint ventures), and sub-units within conservation planning units. (A) Large population objective is set for an individual biological population, with objectives downscaled for conservation units and sub-units. (B) Large population objective is set for the sub-populations in each of several conservation units, with the goal of a resilient biological population. (C) Large population objective is set for the sub-units in each of several conservation units, with the goal of resilient sub-populations within each conservation unit.

for the entire population of the species in Case 1 or Case 3, or each of the individual isolated populations in Case 2; the sizes of each of the populations in Case 2 could not be summed to reach the combined status of *viable* if each one is reproductively isolated.

On the other hand, it may be desirable to apply the framework to smaller segments of a population, such as to meet a conservation goal of maintaining representation of genetic diversity across the population's range, preserving local adaptations, and improving resilience (Tear et al. 2005). To achieve this goal in any of the cases above, a population objective of *viable* or *large* could be set for each of several sub-populations to achieve a *resilient* population overall (Figure 1B). As an example for Case 3, population objectives could be set on the scale of ecoregions (e.g., Olson et al. 2001), conservation regions (e.g., NABCI 2000), or joint ventures (e.g., NAWMP 1998). At a still smaller scale, the managers of any one of these individual conservation planning units could define a conservation goal of achieving multiple genetically robust and self-sustaining sub-populations within their own borders, and set population objectives of *viable* or *large* for each of several subunits to achieve a *resilient* population overall (Figure 1C).

Because conservation plans are often developed on spatial scales that are smaller than the full extent of a biological population, managers may want to apply the framework to individual conservation planning units and sub-units. However, the minimum spatial scale that should be considered depends on the behavior and natural history of the species of interest. For example, estimates of population density or territory size can inform the minimum area required to achieve a *viable* or *large* population. Similarly, estimates of historical population sizes or habitat extent can help estimate the historical capacity of a conservation planning unit or sub-unit to support a *viable* or *large* population. Potential capacity could also be estimated, such as by estimating the historical habitat extent and subtracting the amount permanently lost to urbanization. If a single conservation planning unit or sub-unit is too small in area to support *viable* or *large* populations of the species of interest, it may be more reasonable to redefine the conservation goals to set population objectives for groups of

adjacent planning units or sub-units. For species with particularly low densities or large home ranges, conservation groups may need to coordinate with each other on a common population objective over a large area (Sanderson et al. 2008). Whichever scale is selected, the population objectives can always be downscaled by assigning a percentage of the objective to individual planning units (Figure 1).

For migratory species, another consideration is how to set objectives for non-breeding populations. If there were simple, direct relationships between breeding and non-breeding populations of the same individuals, managers in each area could coordinate on a common population objective to be able to support the population throughout the full annual cycle. More often, migration patterns are complex, with individuals from multiple breeding populations mixing during the non-breeding season and vice versa. In the case of individuals from a single breeding population dispersing to contribute to multiple non-breeding populations, the non-breeding population objective could represent just a fraction of the breeding population objective. However, just as for breeding populations, establishing or maintaining multiple *viable* or *large* non-breeding populations or sub-populations would improve resilience and ecological function during the non-breeding season, and hedge against the risk of environmental catastrophes in one part of the non-breeding season range. In the case of individuals from several separate breeding populations converging to form a single non-breeding population, the non-breeding population objective may need to be much higher to be able to support multiple self-sustaining breeding populations. Further, non-breeding population objectives may also need to reflect within-season movements (e.g., Cormier et al. 2013) and the ebb and flow of individuals during key periods, for example at important migration staging areas (Myers 1983).

The temporal scale on which this framework should be applied is flexible, but we intend population objectives of *viable* or *large* to represent long-term conservation objectives (e.g., 100 years). If the focus is instead on the short-term (e.g., 10 years), it can be difficult to imagine ever achieving *viable* or *large* populations of many species because of current political and economic feasibility, logistical

difficulties, or potential controversy (Manning et al. 2006; Sanderson et al. 2008). Yet there are many examples of successful conservation projects that were once considered impossible (Svancara et al. 2005; Manning et al. 2006). Over the long term, all these circumstances can change, and thus current obstacles should not limit the long-term conservation vision (Sanderson et al. 2008). We recommend viewing population objectives as long-term conservation endpoints at which we expect to reach the goal of secure or robust populations, while short-term milestones or checkpoints can break the long-term objectives down into stages and help progress to be evaluated (Manning et al. 2006; Villard and Jonsson 2009). For example, a short-term population objective may be to move a *very small* population to *small*, to stabilize a population that is *steeply declining*, or simply to maintain the status quo and limit the effects of any new threats.

APPLICATION TO A MULTI-SPECIES REGIONAL CONSERVATION PLAN

We used this population status framework to set long-term population objectives for a suite of breeding riparian bird species in the Central Valley of California, as part of the Central Valley Joint Venture (CVJV), a collaboration of 20 agencies and organizations (Dybala et al. 2017, this volume). Here, we illustrate how this framework can be applied to a multi-species regional conservation plan by briefly outlining the process we undertook to agree on initial long-term population objectives. We then recommend several additional factors to consider in applying this framework to other conservation plans.

First, we identified our long-term conservation goal: for the Central Valley to have riparian ecosystems capable of supporting genetically robust, self-sustaining, ecologically functional, and resilient wildlife populations. We identified a suite of breeding riparian landbird focal species that represented a range of life history characteristics, specific vegetation associations, and current population sizes, which we assumed would be good indicators of the condition of riparian ecosystems (Chase and Geupel 2005). We then examined the current status of each species within each of four planning regions. Although there is likely to be considerable

connectivity among individuals in each planning region, we treated the individuals in each region as though they were separate populations. Synthesizing recent bird survey data across the Central Valley, we estimated the current density of each regional focal species population, and, based on estimates of the current extent of riparian vegetation in the Central Valley, we estimated the current size of each regional population. These ranged from *very small* to *large*, but over half were *very small* or *small*. In contrast, using estimates of the historical extent of riparian vegetation, we estimated that each region easily had the historical capacity to support *viable* or *large* populations of all focal species. Even after accounting for permanent losses of historical riparian vegetation to current and projected urbanization, we estimated that each region still had the potential to support *viable* or *large* populations of all focal species. Thus, we assumed that long-term objectives of *viable* or *large* populations were reasonable conservation endpoints for each region.

With this groundwork laid, we then defined long-term population objectives designed to meet our long-term conservation goal of riparian ecosystems capable of supporting *resilient* wildlife populations within the Central Valley, requiring multiple *viable* or *large* regional populations of each focal species. Therefore, we defined objectives that would meet the following criteria: 1) within each region, all focal species populations are *viable* (>10,000); 2) within each region, most (70%) focal species populations are *large* (>50,000); and 3) across all regions, each focal species has at least one *large* (>50,000) population (Figure 2). We took this approach, rather than setting all *large* objectives, because our primary interest in these species was as indicators of ecosystem condition, and we recognized that some regions were likely better suited to some focal species than others. Thus, we did not expect to be able to achieve *large* populations of all focal species in all regions, but we aimed to ensure each species would have at least one *large* population within the Central Valley.

With these population objectives defined, conservation strategies and management plans for achieving the objectives can be developed, implemented, and evaluated (CMP 2013). In our case, with over 95% of the historical riparian vegetation in the Central Valley lost (Katibah 1984),

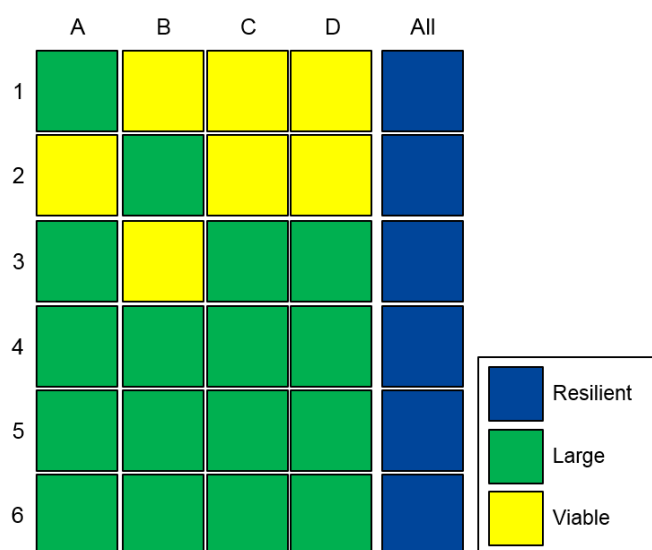


Figure 2 Example of population objectives set for multiple species (1–6) in multiple planning regions (A–D) within a conservation planning unit. These objectives meet the criteria of viable populations for all species within each region, large populations for most species (>70%) within each region, and at least one large regional population for each species. The objectives allow for variation in the suitability of each region for each species. For example, species 3 is less suited to region B and naturally has lower densities there, making it harder to achieve a large population, whereas species 2 naturally has higher densities in region B. Species 1 has low densities everywhere, but it would be easiest to achieve a large population in region A with the most existing habitat. By achieving these objectives, each species would have more than one viable or large regional population, to reach the goal of resilient populations within the larger conservation planning unit.

we assumed that habitat availability is the primary limiting factor for most species. Thus, we also set riparian habitat objectives for each region, based on estimates of how much additional riparian vegetation would be required to meet the population objectives. These habitat objectives are useful estimates of the magnitude of riparian habitat restoration that will be needed, and of how much funding will need to be secured for those projects. Monitoring plans will be essential to measuring the actual contribution of restoration projects toward reaching the long-term population objectives, identifying the most successful practices, and informing any necessary revisions. When habitat quantity may not be the primary limiting factor, region-wide monitoring and/

or specific research effort will be needed to identify other conservation priorities, such as efforts to identify the causes of *steeply declining* populations.

In applying this framework to our system, we identified several considerations that might suggest revisions to the general population status framework when it is applied to other systems:

1. Beyond *large, resilient* populations, are there additional goals or stakeholder interests that should be included in the conservation planning process? For example, if one of the goals was to accommodate recreational hunting, the population objective might need to be substantially higher than the numbers proposed here (Sanderson 2006).
2. Is there any additional information tailored to the individual species or populations of interest that suggests adjustments to the population status thresholds? These might include genetic studies, population viability analyses, or research on ecological function. For example, research on a keystone species may identify the specific population sizes or densities required to maintain the current ecosystem state (Estes et al. 2010).
3. Are any future changes anticipated, such as effects of climate change on the ecosystem or populations of interest? For example, the potential capacity of the planning unit to support a species of interest may change if the distribution of the species is expected to shift into or out of the area, or if sea level rise or changing weather patterns will limit potential habitat (Galbraith et al. 2002; Stralberg et al. 2009). In these cases, the conservation goals, the species of interest, and/or the geographical scale of the conservation plan may require adjustment to accommodate these projected shifts.

Answering these questions is not simple and will require decision-making despite many uncertainties. Thus, we recommend considering the long-term population objectives as hypotheses for the population sizes required to meet the conservation goals. As such, they should be reviewed regularly and refined to incorporate new information (Armstrong and Wittmer 2009).

DISCUSSION

Our framework provides transparent, science-based guidelines for setting population objectives designed to meet long-term conservation goals. The population size thresholds we have proposed represent general hypotheses for the orders of magnitude necessary for most populations to reach each population status. This framework is not a replacement for detailed species- or population-specific genetic, demographic, or population dynamics information, particularly for threatened and endangered species. Where these data are available or can be obtained, they should always be preferred. However, for most species and populations, these data are lacking and would be prohibitively time- and resource-intensive to obtain, especially for entire suites of focal species without special legal status. In these cases, our framework represents a useful set of initial hypotheses, allowing conservation plans to get underway while still allowing the objectives to be further tailored to individual situations and revised as additional information becomes available.

Even with this framework, the process of setting population objectives is not strictly scientific. The appropriate spatial scale at which the framework should be applied will depend on the ultimate conservation goals, the species of interest, and the historical or potential capacity of the conservation planning unit, while the population objectives may require adjustments to reflect additional stakeholder values or anticipated effects of climate change. However, the complexity of the problem makes these guidelines an invaluable starting point for setting long-term population objectives of an appropriate magnitude to achieve conservation goals of genetically robust, self-sustaining, ecologically functional, and resilient populations.

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REFERENCES

- Anderson SH, Kelly D, Ladley JJ, Molloy S, Terry J. 2011. Cascading effects of bird functional extinction reduce pollination and plant density. *Science* 331:1068–1071. <https://doi.org/10.1126/science.1199092>
- Armstrong DP, Wittmer HU. 2009. Setting quantitative targets for recovery of threatened species. In: Villard M-A, Jonsson BG, editors. *Setting Conservation Targets for Managed Forest Landscapes*. Cambridge (UK): Cambridge University Press. p. 264–282. <https://doi.org/10.1017/CBO9781139175388.014>
- Barlow KE, Briggs PA, Haysom KA, Hutson AM, Lechiara NL, Racey PA, Walsh AL, Langton SD. 2015. Citizen science reveals trends in bat populations: The National Bat Monitoring Programme in Great Britain. *Biol Conserv* 182:14–26. <https://doi.org/10.1016/j.biocon.2014.11.022>
- Berryman AA. 2002. Population: A central concept for ecology? *Oikos* 97:439–442. <https://doi.org/10.1034/j.1600-0706.2002.970314.x>
- Brook BW, Bradshaw CJA, Traill LW, Frankham R. 2011. Minimum viable population size: Not magic, but necessary. *Trends Ecol Evol* 26:619–20; author reply 620–2. <https://doi.org/10.1016/j.tree.2011.09.006>
- Chapman AP, Brook BW, Clutton-Brock TH, Grenfell BT, Frankham R. 2001. Population viability analyses on a cycling population: a cautionary tale. *Biol Conserv* 97:61–69. [https://doi.org/10.1016/S0006-3207\(00\)00100-2](https://doi.org/10.1016/S0006-3207(00)00100-2)
- Chase MK, Geupel GR. 2005. The use of avian focal species for conservation planning in California. [cited 2016 Feb 26]. USDA Forest Service. Gen. Tech. Rep. PSW-GTR-191. p. 130–142. Available from: http://www.fs.fed.us/psw/publications/documents/psw_gtr191/psw_gtr191_0130-0142_chase.pdf
- [CMP] Conservation Measures Partnership. 2013. *The open standards for the practice of conservation*. 3rd ed. Washington, D.C.: Conservation Measures Partnership. [cited 2016 Feb 26]. Available from: <http://cmp-openstandards.org/download-os/>
- Conner RN. 1988. Wildlife populations: minimally viable or ecologically functional? *Wildl Soc Bull* 16:80–84. Available from: <http://www.jstor.org/stable/3782359>

- Cormier RL, Humple DL, Gardali T, Seavy NE. 2013. Light-level geolocators reveal strong migratory connectivity and within-winter movements for a coastal California Swainson's Thrush (*Catharus ustulatus*) population. *Auk* 130:283–290. <https://doi.org/10.1525/auk.2013.12228>
- Doak DF, Himes Boor GK, Bakker VJ, Morris WF, Louthan A, Morrison SA, Stanley A, Crowder LB, Stanley A, Crowder LB. 2015. Recommendations for improving recovery criteria under the US Endangered Species Act. *Bioscience* 65:189–199. <https://doi.org/10.1093/biosci/biu215>
- Dybala KE, Clipperton N, Gardali T, Golet G, Lorenzato S, Melcer R, Seavy N, Silveira J, Yarris G. 2017. Population and habitat objectives for avian conservation in California's Central Valley riparian ecosystems. *San Franc Estuary Watershed Sci* 15(1). <https://doi.org/10.15447/sfews.2017v15iss1art5>
- Estes JA, Tinker MT, Bodkin JL. 2010. Using ecological function to develop recovery criteria for depleted species: sea otters and kelp forests in the Aleutian Archipelago. *Conserv Biol* 24:852–860. <https://doi.org/10.1111/j.1523-1739.2009.01428.x>
- Flather CH, Hayward GD, Beissinger SR, Stephens PA. 2011. Minimum viable populations: Is there a “magic number” for conservation practitioners? *Trends Ecol Evol* 26:307–316. <https://doi.org/10.1016/j.tree.2011.03.001>
- Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CSS. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Annu Rev Ecol Evol Syst* 35:557–581. <https://dx.doi.org/10.1146/annurev.ecolsys.35.021103.105711>
- Frankham R. 1995. Effective population size/adult population size ratios in wildlife: a review. *Genet Res* 66:95–107. <https://doi.org/10.1017/S0016672308009695>
- Frankham R, Bradshaw CJA, Brook BW. 2014. Genetics in conservation management: revised recommendations for the 50/500 rules, Red List criteria and population viability analyses. *Biol Conserv* 170:56–63. <https://doi.org/10.1016/j.biocon.2013.12.036>
- Franklin IR. 1980. Evolutionary change in small populations. In: Soulé M, Wilcox B, editors. *Conservation biology: an evolutionary-ecological perspective*. Sunderland (MA): Sinauer Associates. p. 135–149.
- Galbraith H, Jones R, Park R, Clough J, Herrod-Julius S, Harrington B, Page G. 2002. Global climate change and sea level rise: potential losses of intertidal habitat for shorebirds. *Waterbirds* 25:173–183. [https://doi.org/10.1675/1524-4695\(2002\)025\[0173:GCCASL\]2.0.CO;2](https://doi.org/10.1675/1524-4695(2002)025[0173:GCCASL]2.0.CO;2)
- Galetti M, Guevara R, Côrtes MC, Fadini R, Von Matter S, Leite AB, Labecca F, Ribeiro T, Carvalho CS, Collevatti RG, et al. 2013. Functional extinction of birds drives rapid evolutionary changes in seed size. *Science* 340:1086–1090. <https://doi.org/10.1126/science.1233774>
- Himes Boor GK. 2014. A framework for developing objective and measurable recovery criteria for threatened and endangered species. *Conserv Biol* 28:33–43. <https://doi.org/10.1111/cobi.12155>
- Iliff M, Salas L, Inzunza ER, Ballard G, Lepage D, Kelling S. 2009. The Avian Knowledge Network: a partnership to organize, analyze, and visualize bird observation data for education, conservation, research, and land management. [cited 2016 Feb 26]. In: *Tundra to tropics: connecting birds, habitats and people*. Proceedings of the 4th International Partners in Flight Conference; 13–16 Feb 2008, McAllen, TX. p. 365–373. Available from: [https://www.fs.fed.us/psw/publications/4251/psw_2009_salas\(iliff\)001.pdf](https://www.fs.fed.us/psw/publications/4251/psw_2009_salas(iliff)001.pdf)
- Jamieson IG, Allendorf FW. 2012. How does the 50/500 rule apply to MVPs? *Trends Ecol Evol* 27:578–584. <https://doi.org/10.1016/j.tree.2012.07.001>
- Katibah EF. 1984. A brief history of riparian forests in the Central Valley of California. In: Warner RE, Hendrix KM, editors. *California riparian systems: ecology, conservation, and productive management*. [cited 2016 Feb 26]. Berkeley (CA): University of California Press. Available from: <http://ark.cdlib.org/ark:/13030/ft1c6003wp/>
- Lawler JJ. 2009. Climate change adaptation strategies for resource management and conservation planning. *Ann NY Acad Sci* 1162:79–98. <https://doi.org/10.1111/j.1749-6632.2009.04147.x>

- [LCC] Landscape Conservation Cooperative. 2014. Network strategic plan. [cited 2016 Feb 26]. Available from: <https://lccnetwork.org/resource/landscape-conservation-cooperative-network-strategic-plan>
- Mace GM, Collar NJ, Gaston KJ, Hilton-Taylor C, Akçakaya HR, Leader-Williams N, Milner-Gulland EJ, Stuart SN. 2008. Quantification of extinction risk: IUCN's system for classifying threatened species. *Conserv Biol* 22:1424–1442. <https://doi.org/10.1111/j.1523-1739.2008.01044.x>
- Manning AD, Lindenmayer DB, Fischer J. 2006. Stretch goals and backcasting: approaches for overcoming barriers to large-scale ecological restoration. *Restor Ecol* 14:487–492. <https://doi.org/10.1111/j.1526-100X.2006.00159.x>
- Margules CR, Pressey RL. 2000. Systematic conservation planning. *Nature* 405:243–253. <https://doi.org/10.1038/35012251>
- Marsh H, De'ath G, Gribble N, Lane B. 2005. Historical marine population estimates: triggers or targets for conservation? The dugong case study. *Ecol Appl* 15:481–492. <https://doi.org/10.1890/04-0673>
- McClenachan L, Ferretti F, Baum JK. 2012. From archives to conservation: Why historical data are needed to set baselines for marine animals and ecosystems. *Conserv Lett* 5:349–359. <https://doi.org/10.1111/j.1755-263X.2012.00253.x>
- Meek MH, Wells C, Tomalty KM, Ashander J, Cole EM, Gille DA, Putman BJ, Rose JP, Savoca MS, Yamane L, et al. 2015. Fear of failure in conservation: the problem and potential solutions to aid conservation of extremely small populations. *Biol Conserv* 184:209–217. <https://doi.org/10.1016/j.biocon.2015.01.025>
- Michener WK, Allard S, Budden A, Cook RB, Douglass K, Frame M, Kelling S, Koskela R, Tenopir C, Vieglais DA. 2012. Participatory design of DataONE-Enabling cyberinfrastructure for the biological and environmental sciences. *Ecol Inform* 11:5–15. <https://doi.org/10.1016/j.ecoinf.2011.08.007>
- Mills LS, Scott JM, Strickler KM, Temple SA. 2005. Ecology and management of small populations. In: Braun CE, editor. *Techniques for wildlife investigations and management*. [cited 2016 Feb 26]. Bethesda (MD): The Wildlife Society. p. 691–713. Available from: http://media.longnow.org/files/2/REVIVE/chap35sm_Part_1.pdf
- Morris WF, Bloch PL, Hudgens BR, Moyle LC, Stinchcombe JR. 2002. Population viability analysis in endangered species recovery plans: past use and future improvements. *Ecol Appl* 12:708–712. [https://doi.org/10.1890/1051-0761\(2002\)012\[0708:PVAIES\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2002)012[0708:PVAIES]2.0.CO;2)
- Morris WF, Doak DF. 2002. *Quantitative conservation biology*. Sunderland (MA): Sinauer Associates, Inc.
- Myers JP. 1983. Conservation of migrating shorebirds: staging areas, geographic bottlenecks, and regional movements. *Am Birds* 37:23–25. Available from: <https://sora.unm.edu/node/114172>
- [NABCI] North American Bird Conservation Initiative. 2000. Bird conservation region descriptions. [cited 2016 Feb 26]. Arlington (VA): NABCI. Available from: <http://nabci-us.org/resources/bird-conservation-regions-map/>
- [NAWMP] North American Waterfowl Management Plan. 1986. North American Waterfowl Management Plan: a strategy for cooperation. Available from: <https://www.fws.gov/migratorybirds/pdf/management/NAWMP/1998NAWMP.pdf>
- [NAWMP] North American Waterfowl Management Plan. 1998. Expanding the vision: 1998 update. Available from: <https://www.fws.gov/migratorybirds/pdf/management/NAWMP/OriginalNAWMP.pdf>
- Neel MC, Leidner AK, Haines A, Goble DD, Scott JM. 2012. By the numbers: how is recovery defined by the US Endangered Species Act? *Bioscience* 62:646–657. <https://doi.org/10.1525/bio.2012.62.7.7>
- Nicholson E, Possingham HP. 2006. Objectives for multiple-species conservation planning. *Conserv. Biol.* 20:871–881. <https://doi.org/10.1111/j.1523-1739.2006.00369.x>
- [NOAA] National Oceanic and Atmospheric Administration. 2012. International marine mammal action plan 2012–2016. [cited 2016 Feb 26]. Silver Spring (MD): NOAA. Available from: http://www.nmfs.noaa.gov/ia/species/marine_mammals/immap.pdf
- Noss RF, Dobson AP, Baldwin R, Beier P, Davis CR, Dellasala DA, Francis J, Locke H, Nowak K, Lopez R, et al. 2012. Bolder thinking for conservation. *Conserv Biol* 26:1–4. <https://doi.org/10.1111/j.1523-1739.2011.01738.x>

- Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GVN, Underwood EC, D'Amico JA, Itoua I, Strand HE, Morrison JC, et al. 2001. Terrestrial ecoregions of the world: A new map of life on Earth. *Bioscience* 51:933. Available from: <https://academic.oup.com/bioscience/article/51/11/933/227116/Terrestrial-Ecoregions-of-the-World-A-New-Map-of>
- Patterson BR, Murray DL. 2008. Flawed population viability analysis can result in misleading population assessment: A case study for wolves in Algonquin park, Canada. *Biol Conserv* 141:669–680. <https://doi.org/10.1016/j.biocon.2007.12.010>
- Peery CA, Kavanagh KL, Scott JM. 2003. Pacific salmon: setting ecologically defensible recovery goals. *Bioscience* 53:622–623. Available from: <https://academic.oup.com/bioscience/article/53/7/622/219775/Pacific-Salmon-Setting-Ecologically-Defensible>
- Petrie MJ, Brasher MG, Soulliere GJ, Tirpak JM, Pool DB, Reker RR. 2011. Guidelines for establishing Joint Venture waterfowl population abundance objectives. North American Waterfowl Management Plan Science Support Team, Tech. Rpt. No. 2011-1. Available from: https://www.fws.gov/migratorybirds/pdf/management/NAWMP/GuidelinesforEstablishing%20JVPopulationObjectives%20FinalReport_draft_alt_format.pdf
- Pyare S, Berger J. 2003. Beyond demography and delisting: Ecological recovery for Yellowstone's grizzly bears and wolves. *Biol Conserv* 113:63–73. [https://doi.org/10.1016/S0006-3207\(02\)00350-6](https://doi.org/10.1016/S0006-3207(02)00350-6)
- Redford KH, Amato G, Baillie J, Beldomenico P, Bennett EL, Clum N, Cook R, Fonseca G, Hedges S, Launay F, et al. 2011. What does it mean to successfully conserve a (vertebrate) species? *Bioscience* 61:39–48. <https://doi.org/10.1525/bio.2011.61.1.9>
- Reed DH, O'Grady JJ, Brook BW, Ballou JD, Frankham R. 2003. Estimates of minimum viable population sizes for vertebrates and factors influencing those estimates. *Biol Conserv* 113:23–34. [https://doi.org/10.1016/S0006-3207\(02\)00346-4](https://doi.org/10.1016/S0006-3207(02)00346-4)
- Rich TD, Beardmore CJ, Berlanga H, Blancher PJ, Bradstreet MSW, Butcher GS, Demarest DW, Dunn EH, Hunter WC, Iñigo-Elias EE, et al. 2004. Partners in Flight North American landbird conservation plan. [cited 2016 Feb 26]. Ithaca (NY): Cornell Lab of Ornithology. Available from: <https://www.partnersinflight.org/wp-content/uploads/2016/07/PIF-Landbird-Conservation-Plan-2004.pdf>
- Ripple WJ, Wirsing AJ, Wilmsers CC, Letnic M. 2013. Widespread mesopredator effects after wolf extirpation. *Biol Conserv* 160:70–79. <https://doi.org/10.1016/j.biocon.2012.12.033>
- Royle JA, Nichols JD. 2003. Estimating abundance from repeated presence-absence data or point counts. *Ecology* 84:777–790. [https://doi.org/10.1890/0012-9658\(2003\)084\[0777:EAFRPA\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2003)084[0777:EAFRPA]2.0.CO;2)
- Sanderson EW. 2006. How many animals do we want to save? The many ways of setting population target levels for conservation. *Bioscience* 56:911–922. Available from: <https://academic.oup.com/bioscience/article/56/11/911/272329/How-Many-Animals-Do-We-Want-to-Save-The-Many-Ways>
- Sanderson EW, Redford KH, Weber B, Aune K, Baldes D, Berger J, Carter D, Curtin C, Derr J, Dobrott S, et al. 2008. The ecological future of the North American bison: Conceiving long-term, large-scale conservation of wildlife. *Conserv Biol* 22:252–266. <https://doi.org/10.1111/j.1523-1739.2008.00899.x>
- Soulé ME, Estes JA, Berger J, Del Rio CM. 2003. Ecological effectiveness: Conservation goals for interactive species. *Conserv Biol* 17:1238–1250. <https://doi.org/10.1046/j.1523-1739.2003.01599.x>
- Stanton JC. 2014. Present-day risk assessment would have predicted the extinction of the passenger pigeon (*Ectopistes migratorius*). *Biol Conserv* 180:11–20. <https://doi.org/10.1016/j.biocon.2014.09.023>
- Stralberg D, Jongsomjit D, Howell CA, Snyder MA, Alexander JD, Wiens JA, Root TL. 2009. Re-shuffling of species with climate disruption: a no-analog future for California birds? *PLoS One* 4:e6825. <https://doi.org/10.1371/journal.pone.0006825>
- Sullivan BL, Aycrigg JL, Barry JH, Bonney RE, Bruns N, Cooper CB, Damoulas T, Dhondt AA, Dietterich T, Farnsworth A, et al. 2014. The eBird enterprise: an integrated approach to development and application of citizen science. *Biol Conserv* 169:31–40. <https://doi.org/10.1016/j.biocon.2013.11.003>
- Svancara LK, Brannon R, Scott JM, Groves CR, Noss RF, Pressey RL. 2005. Policy-driven versus evidence-based conservation: a review of political targets and biological needs. *Bioscience* 55:989–995. Available from: <https://academic.oup.com/bioscience/article/55/11/989/220923/Policy-driven-versus-Evidence-based-Conservation-A>

- Tear TH, Kareiva P, Angermeier PL, Comer P, Czech B, Kautz R, Landon L, Mehlman D, Murphy K, Ruckelshaus M, et al. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. *Bioscience* 55:835–849. Available from: <https://academic.oup.com/bioscience/article/55/10/835/274365/How-Much-Is-Enough-The-Recurrent-Problem-of>
- Trall L, Bradshaw C, Brook B. 2007. Minimum viable population size: A meta-analysis of 30 years of published estimates. *Biol Conserv* 139:159–166. <https://doi.org/10.1016/j.biocon.2007.06.011>
- Trall LW, Brook BW, Frankham RR, Bradshaw CJA. 2010. Pragmatic population viability targets in a rapidly changing world. *Biol Conserv* 143:28–34. <https://doi.org/10.1016/j.biocon.2009.09.001>
- [UNEP] United Nations Environment Programme. 2010. Strategic plan for biodiversity 2011–2020 and the Aichi biodiversity targets. In: Conference of the Parties to the Convention on Biological Diversity; 18–29 October 2010, Nagoya, Japan.
- Villard M-A, Jonsson BG. 2009. A plea for quantitative targets in biodiversity conservation. In: Villard M-A, Jonsson BG, editors. *Setting conservation targets for managed forest landscapes*. [cited 2016 Feb 26]. Cambridge (UK): Cambridge University Press. p. 1–8. Available from: http://assets.cambridge.org/9780521877091/excerpt/9780521877091_excerpt.pdf
- Westwood A, Reuchlin-Hugenholtz E, Keith DM. 2014. Re-defining recovery: a generalized framework for assessing species recovery. *Biol Conserv* 172:155–162. <https://doi.org/10.1016/j.biocon.2014.02.031>
- Wilhere GF. 2008. The how-much-is-enough myth. *Conserv Biol* 22:514–517. <https://doi.org/10.1111/j.1523-1739.2008.00926.x>
- Wolf S, Hartl B, Carroll C, Neel MC, Greenwald DN. 2015. Beyond PVA: why recovery under the Endangered Species Act is more than population viability. *Bioscience* 65:200–207. <https://doi.org/10.1093/biosci/biu218>